CHAPTER

19

Human Activity and Habitat Loss: Destruction, Fragmentation, and Degradation

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19.1 IMPACT OF AGRICULTURE

19.1.1 Overview

The major form of habitat destruction is deforestation either to develop land for agriculture (70%) or to harvest lumber intensively (World Wildlife Fund, 2015). Two examples of habitat destruction via deforestation will be considered here: (1) deforestation in the Amazon Basin and (2) deforestation in Malaysia, Indonesia, and sub-Saharan Africa from replacement of the natural habitat with oil palm (*Elaeis guineensis*) plantations.

19.1.2 Habitat Destruction and Deforestation in the Amazon Basin

There has been economic pressure to convert the forests of the Amazon to pasture and arable land (Nepstad et al., 2008). The pressures that increase land values include the following (Nepstad et al., 2008):

- Growth of sugarcane for ethanol production as an alternative energy source.
- Growth of palm oil for biodiesel—another biorenewable energy source.
- Corn and soybean production; there being a ready market for corn and soybeans particularly for the growing pig and poultry sector in Brazil but also for exports.
- Pasture for beef cattle.

However, between 2004 and 2010, the rate of deforestation declined by 70% due to the implementation of government policies (Macedo et al., 2012).

Deforestation can be related to other human activities including logging and fires (e.g., in slash and burn subsistence farming) together with such indirect effects as drought (a consequence of decreased rainfall following loss of forests) (Nepstad et al., 2008). The World Wildlife Fund (2015) claims that "Extensive cattle ranching is the number one culprit of deforestation in virtually every Amazon country, and it

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accounts for 80% of current deforestation." In contrast, refereed quantitative research demonstrates that the major increases in soy production in the Amazon between 2001 and 2010 have been predominantly due to soybean farming on previously cleared pasture areas (Macedo et al., 2012).

19.1.3 Loss of Forests to Oil Palm Plantations in Malaysia, Indonesia, and Sub-Saharan Africa

Palm oil makes up 32% of the global supply of fats and oils (cf. 22% soybean oil) (Palm Oil Research, 2017). Palm oil is used in frying foods including in the fast food industry and in food and nonfood manufacturing. The basis for the increase in the production of palm oil is its production efficiency in comparison with other vegetable oil sources, such as rapeseed and soybean with 1 mt of oil being produced from the following land areas (Palm Oil Research, 2017):

- palm oil from 0.26 ha
- rapeseed oil from 1.5 ha
- soybean oil from 2.2 ha

Production of palm oil is growing (Table 19.1). Native forests have been converted to oil palm plantations in Malaysia and Indonesia. The oil palm originated in West Africa (FAO, 2005). According to Palm Oil Research (2017), 14.2 million ha (35.1 million acres) are used globally for the production of palm oil (compared to 258.9 million ha for agriculture). Today, there are many oil palm plantations particularly in Malaysia and Indonesia. A measure of the importance of oil palms in Malaysia is that 71% of agricultural land is used for palm oil production (Palm Oil Research, 2017). Large oil palm farms can be >500 ha.

Given the increase in palm oil production in Indonesia (Table 19.1), it is not surprising that deforestation is occurring at an accelerating rate (Wich et al., 2011). The Union of Concerned Scientists (Union of Concerned Scientists, in press) argue that the deforestation is having adverse effects on endangered species (Fig. 19.1), such as Bornean orangutans, Sumatran orangutans, pygmy elephants, and tigers and contributing to climate change with the release of carbon dioxide. It has been estimated that "49% of the orangutan distribution will be lost if all forests outside of protected areas and logging concessions are lost" (Wich et al., 2012). There is increasing palm oil production in Africa and with it concerns on the impact on great ape populations and other threatened species (Wich et al., 2014). This is particularly important given the substantial overlap (~60%) between the current oil palm concessions and the range of the great apes in Africa (Wich et al., 2014). There are parallel grave concerns from environmental and conservation groups not only about losses of biodiversity including the great apes and the release of greenhouse gases but also for the indigenous peoples displaced

TABLE 19.1 Increase in Palm Kernel Production in Million Metric Tons (FAOStat, 2015) Destroys Habitats for Animals

	Palm kennel production (million metric tons)		
Year	**World**	**Malaysia**	**Indonesia**
1993	4.44	2.27	0.82
2003	8.25	3.63	2.48
2013	15.64	4.86	7.53

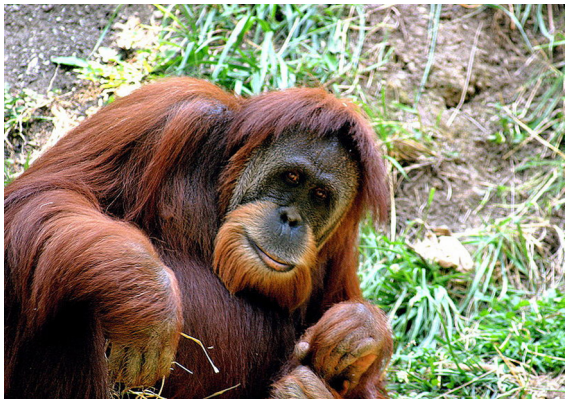

FIGURE 19.1 **Orangutan.** Habitat of orangutans is being destroyed with the land going to palm oil plantations. *Source: Image from Shutterstock.*

with the development of oil palm plantations. Such groups include Friends of the Earth (2005), Greenpeace (2014), and zoos. The latter is exemplified by the zoo-based educational campaigns successfully developed at the Melbourne Zoo, Australia with 160,000 people signing an associated petition (Pearson et al., 2014).

19.2 IMPACT OF URBANIZATION

The number and proportion of people living in urban areas are increasing rapidly (Table 19.2). Between 1970 and 2000, there was an increase in urban land area of 5.8 million ha globally (Seto et al., 2011). Between 2000 and 2010, 2.8 million ha was urbanized in China alone (World Bank Group, 2015). The terms urbanization and urban areas are not necessarily consistently employed with an urban area frequently based on administrative boundaries (Liu et al., 2014). Urban areas include parks, yards/gardens, medians on highways, and golf-courses. Another method of quantitating urbanization includes "built-up areas" or "place dominated by the built environment" or "impervious surfaces" (Liu et al., 2014) (Fig. 19.2). Tables 19.3 and 19.4 summarize the present global situation for urban areas, built-up areas, and impervious surfaces globally and in major regions/continents. Akin to urbanization is the increase in human populations close to or in protected areas. Table 19.5 summarizes

TABLE 19.2 The Global Urban Population is Dramatically Increasing

	Urban population (billion)	Percentage of world's population
1950	0.7	29.0
1975	1.5	37.2
2000	2.8	46.7
2030 (projected)	4.9	59.9

Data from UN Department of Economic and Social Affairs Population Division), 2005. World urban population. Available from: http://www.un.org/esa/population/publications/WUP2005/2005WUP_FS1.pdf

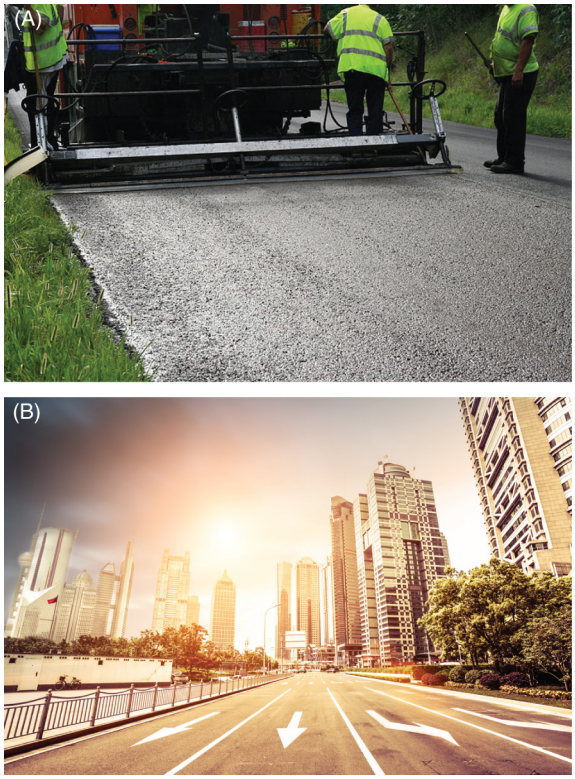

FIGURE 19.2 **Impervious surfaces in a city destroying habitat and/or degrading habitat and/or fragmenting habitats.** *Source: Image from Shutterstock.*

TABLE 19.3 Contribution of Urban, Built-Up, and Impervious Areas to Global Land Use

Land use	Percent of global land area[a]	Land area, ha (km^2)
Urban areas	3	0.3 billion (3 million)
Built-up areas	0.65	75 million (0.7 million)
Impervious surface (roads, buildings, parking lots, etc.)	0.45	60 million (0.6 million)

a Excluding Greenland and Antarctica.
Data from Liu, Z., He, C., Zhou, Y., Wu, J., 2014. How much of the world's land has been urbanized, really? A hierarchical framework for avoiding confusion. Landscape Ecol. 29, 763–771.

TABLE 19.4 Differences in Urban, Built-Up, and Impervious Areas to Land Use in Different Regions of the World

	Area (km^2)		
Region	**Urban areas**	**Built-up areas**	**Impervious surfaces**
Africa	0.8	0.2	0.1
Asia	3.9	0.9	0.6
Europe	3.8	0.9	0.7
Latin America + Caribbean	2.3	0.6	0.3
North America	4.7	0.9	0.8
Oceania	0.6	0.2	0.05

Data from Liu, Z., He, C., Zhou, Y., Wu, J., 2014. How much of the world's land has been urbanized, really? A hierarchical framework for avoiding confusion. Landscape Ecol. 29, 763–771.

TABLE 19.5 Increases in the Number of Housing Units Close to Areas in the USA With High or Relatively High Governmental Protection of Ecosystems

	Area (km^2)	**Housing units within 50 km in 2000 (compared to 1940)**	**Housing units within 1 km in 2000 (compared to 1940)**
Wilderness (highest level of protection)	191,000	20.5 million (4.4 million)	54,000 (9,400)
National parks (managed for ecosystem protection + recreation)	103,000	6.6 million (1.5 million)	85,000
National forests (managed for sustainable use)	869,000	34.8 million (9.0 million)	1.8 million (0.48 million)

Data from Radeloff, V.C., Stewart, S.I., Hawbaker, T.J., Gimmi, U., Pidgeon, A.M., Flather, C.H., Hammer, R.B., Helmers, D.P., 2010. Housing growth in and near United States protected areas limits their conservation value. Proc. Natl. Acad. Sci. USA 107, 940–945.

the increases in housing units and hence, people close to protected areas in the USA.

Urbanization is one of the primary causes of habitat loss and degradation globally (Fig. 19.3). It has severe impacts on the environment with the following (Grimm et al., 2008; Radeloff et al., 2010; Seto et al., 2011):

- Shifts in land use and cover and consequent habitat loss and fragmentation and hence effects on biodiversity.
- Impacts on hydrological systems including destroyed or disrupted wetlands.
- Polluted air and waterways.
- Effects on local and regional climate.
- Increased predation by mesopredators, pets, and pest rodents.

FIGURE 19.3 Land being cleared for housing development destroying or degrading habitats. *Source: Image from Shutterstock.*

19.3 IMPACT OF ROADS ON HABITAT LOSS

19.3.1 Overview

In 2010, there were 25.9 million miles (41.7 million km) of roads globally (Laurance et al., 2014). It is estimated that by 2050, there will be an additional 15.5 million miles (25 million km) of roads with over 90% being in developing countries (Laurance et al., 2014) (Fig. 19.3). New roads are associated with the following negative environmental impacts (Laurance et al., 2014):

- habitat loss (e.g., destruction of wetlands)
- habitat fragmentation
- habitat degradation (e.g., polluted land, water, or air and increased access by human hunters and invasive species)

The effects of roads were categorized by Spellerberg (1998) as follows:

1. Impact of the construction process.
2. Short- and long-term effects (mortalities; increases in carrion-consuming animals; disruption through noise; pollution in run-off, such as lead, cadmium, nickel, and zinc; and emissions including carbon monoxide and NO_x) and increased access of invasive species and humans together with the adverse effects on aquatic environments—adjacent and downstream.
3. Effects of structures associated with roads (clearing rights-of-way), bridges, tunnels, and so on.

Road construction can lead to loss of biodiversity locally or regionally with local extinctions of susceptible species particularly in wetlands with (or without) recolonization (Findlay and Bourdages, 2000).

Roads influence bird populations with the greatest effects being vehicle-caused road mortality and indirect effects of traffic noise (Kociolek et al., 2011). In addition, there are direct effects, such as habitat fragmentation, pollution, and poisoning together with indirect effects, such as roadway lighting, barriers to movement, and edges (Kociolek et al., 2011). The number and regular breeding of grassland birds is decreased close to roads with the effect greater near roads with heavy traffic (Forman et al., 2002). Similarly, there are lower population densities for stone curlews (*Burhinus oedicnemus*) either relatively close to major roads (1000 m) or in settlements (1500 m) (Clarke et al., 2013).

Road substrate and/or traffic also influences the movement of small mammals and reptiles with many species avoiding paved rural two-lane roads but not dirt and secondary paved roads; small mammals [San Diego pocket mouse (*Chaetodipus fallax*), cactus mouse (*Peromyscus eremicus*), Dulzura kangaroo rat (*Dipodomys simulans*), and deer mouse (*Peromyscus maniculatus*)] and reptiles [western fence lizard (*Sceloporus occidentalis*) and orange-throated whiptail (*Aspidoscelis hyperythra*)] all avoiding paved rural two-lane highways (Brehme et al., 2013).

It is widely held that the mortality of turtles on roads is one of the major causes of decline in turtle populations. This is likely to be a fallacy. There was no difference detected in the population density of painted turtles (*Chrysemys picta*) either very close to or some distance from high-traffic roads (Dorland et al., 2014).

19.3.2 Vehicle Wildlife Collisions

Vehicle wildlife collisions have high impacts with the following reported for deer in the USA in 2014 (including elk and moose) according to State Farm (2015):

- 1.2 million collisions (with likely a similar number of deer fatalities)
- $4 billion in vehicle damage.

The most likely months for these collisions are the mating and hunting seasons (October, November, and December) (State Farm, 2015) together with spring fawning (May and June) when yearlings move to new ranges (WI DNR, 2015).

19.4 IMPACT OF THE ENERGY SECTOR

19.4.1 Impact of Oil, Bitumen, and Gas Extraction

19.4.1.1 Mortalities and Toxicity of Petroleum Oil

Marine birds are exposed to oil following spillages. This is seen in studies using mallard ducks as models for water birds. Petroleum oil decreases egg-laying in ducks and also reduces the fertility of the eggs in mallard ducks (Holmes et al., 1978). Unexpectedly, ducks consuming petroleum oil have reduced circulating concentrations of the stress hormone, corticosterone; this being due to increased catabolism (Gorsline and Holmes, 1981). Thus, not only do oil spills kill birds but also are likely to suppress bird populations due to decreased fertility.

19.4.1.2 Impacts of Extraction and Transportation

The development of extraction and transportation for oil, bitumen, and natural gas requires facilities and wells for extraction, pipelines, and railroads for the transportation, and roads to service the system (reviewed by Muhly et al., 2015). The impacts of oil, gas, and bitumen extraction include the following:

- catastrophic failure, such as explosions
- chronic environmental contamination
- habitat loss and fragmentation

19.4.1.3 Catastrophic Failure and Oil, Bitumen, and Natural Gas Extraction

Catastrophic failure of fossil energy extraction and processing systems can lead to deaths of people and wildlife. For example, the explosion of a train carrying oil in the Lac-Mégantic rail disaster in Quebec resulted in 47 people killed and a 1-km blast radius. The fire in the Magurcherra gas field in Bangladesh (1997) lasted for 3 months destroying 60 ha of natural forest with mortalities including deer, monkeys, and birds (Alam et al., 2010). Moreover, explosions or fires can be followed by the release of oil or bitumen into the environment as in the case of the British Petroleum (BP)-leased Deepwater Horizon offshore oil rig (considered later in more detail). Another disaster as the rupture of the oil tanker *Exxon Valdez*. According to the National Oceanic and Atmospheric Administration (NOAA, 1992), the grounded oil tanker *Exxon Valdez* ruptured releasing about 44 million liters of crude oil into Prince William Sound (Alaska) in 1989 with consequent effects on sea otters, birds, and other marine animal life (Harwell et al., 2012). Other major oil spill disasters occurred with the tankers Torrey Canyon and the Amoco Cadiz.

In the spring of 2010, there was a series of catastrophic incidents including an explosion on the BP-leased Deepwater Horizon offshore oil rig in the Gulf of Mexico (Fig. 19.4) (Levy and Gopalakrishnan, 2010). This resulted in the following:

- The release of 0.76 billion liters of crude oil (Goodbody-Gringley et al., 2013).
- The use of 7 million liters of the chemical dispersant Corexit 9500 (Almeda et al., 2014).

This has major effects on deep and coastal marine ecosystems in the Gulf of Mexico together with the 1600 km (1000 miles) of coastline and threats to wetlands (Fig. 19.4). Components of oil spilt in oceans can undergo evaporation, dissolution, dispersion, emulsification, photooxidation, biodegradation, and sedimentation (Rogowska and Namieśnik, 2010).

There were direct and immediate effects with the deaths of dolphins, turtles, and birds and threats to endangered species including the manatee and brown pelicans (Levy and Gopalakrishnan, 2010). In laboratory studies simulating the contamination, weathered oil greatly increased the mortality of coral larvae (*Porites astreoides* and *Montastraea faveolata*) (Goodbody-Gringley et al., 2013). Bioaccumulation of petroleum

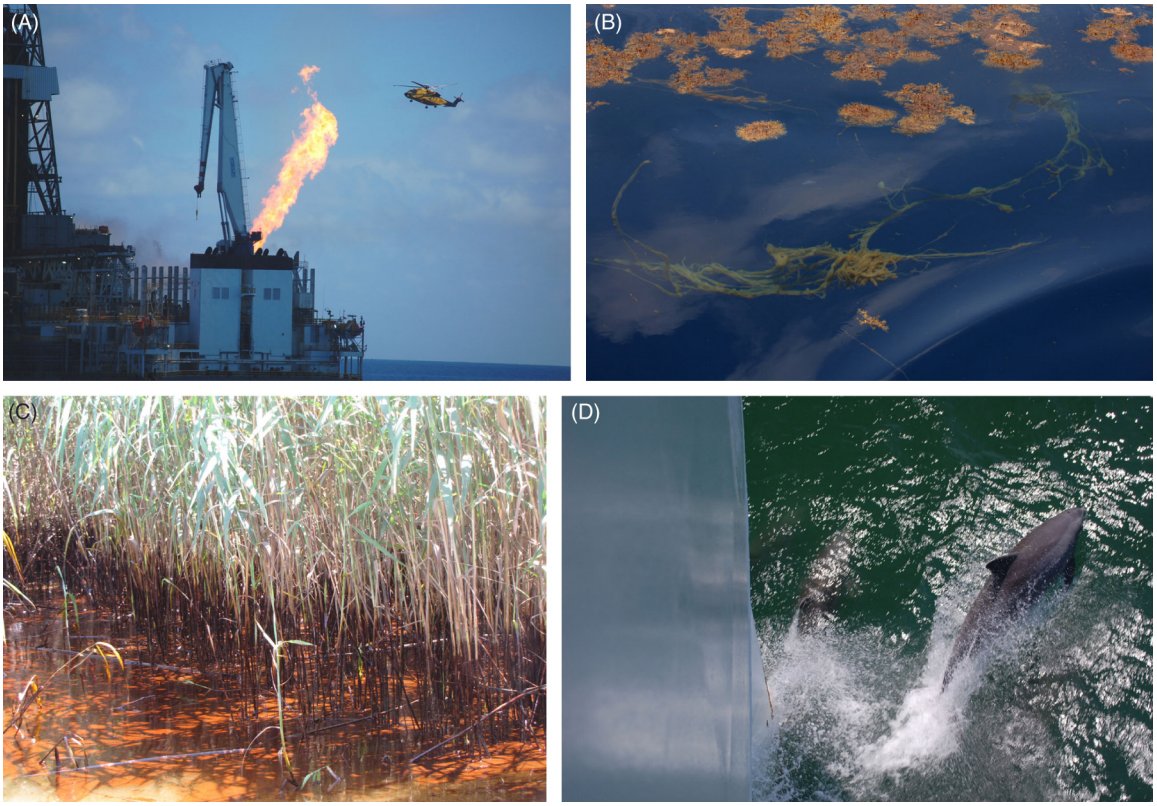

FIGURE 19.4 **An explosion on the BP-leased deepwater horizon offshore oil rig in the Gulf of Mexico resulted in the release of 0.76 billion liters of crude oil and consequent degradation of habitats.** (A) The Deepwater Horizon site. (B) Oil-polluted marine animals including turtles normally associate with sargassum. This sea plant moves with currents along with oil slicks. (C) Massive oil spill resulting in oil emulsion on vegetation in Louisiana, USA. (D) A pod of dolphins following NOAA ship *Pisces* in the Gulf of Mexico after the oil spill. *Source: Part A–D: Courtesy NOAA; part B: photo Carolyn Cole/LA Times.*

hydrocarbons has been demonstrated in fiddler crabs (*Uca minax*) (Chase et al., 2013). In view of the potential for oil to influence nesting birds, it is interesting to note that weathered Gulf oil is toxic to mallard duck embryos but less so than nonweathered oil (Finch et al., 2011). Presumably ducks are more likely to come into contact with weathered oil.

The chemical dispersant Corexit 9500A is highly toxic greatly increasing the mortality of microzooplankton (Almeda et al., 2014) and of coral larvae (Goodbody-Gringley et al., 2013). Moreover, Corexit 9500A increases the toxicity of oil on mallard duck embryos (Finch et al., 2012).

19.4.1.4 Chronic Environmental Contamination

Environmental contamination associated with extractive industries poses risks to wildlife and is viewed as potential habitat degradation. Wildlife is particularly adversely affected when fluids are released, such as the following:

- Oil or bitumen directly released into environment.
- Brine associated with oil and natural gas extraction released into groundwater and hence waterways. These brines have elevated concentrations of chloride with

levels above those suitable for human consumption (Hudak and Wachal, 2001).
- Fracking fluids related to hydraulic fracturing (Vengosha et al., 2013).
- Drilling mud.
- Volatile organic compounds.

The magnitude of the issue is exemplified by the following report about one US state. There were almost 13,000 spills or releases of fluids from oil and gas sites in Oklahoma over 10 years with a median 10 barrels of oil released in each spill or release (Fisher and Sublette, 2005). Leakage can come from active or inactive wells; the latter respective of whether unplugged or inadequately plugged wells or monitored or abandoned. Potential risks from the rupture of oil pipelines also include contamination of wetlands and watersheds (Service et al., 2012).

19.4.1.5 Habitat Loss and Fragmentation From the Development of Oil, Gas, and Bitumen

The ecotype boreal caribou (*Rangifer tarandus caribou*) is threatened and is rapidly declining in Alberta. This has been linked to the development of the Fayetteville shale field for natural gas and the associated increases in roads and edge habitats (Moran et al., 2015). Parenthetically, studies show the caribou crossing under pipelines (Muhly et al., 2015).

19.4.2 Coal Mining

There is a lack of literature quantifying the mortalities or displacement of wildlife by coal mining in the USA (Buehler and Percy, 2002). That is not to say there is no impact. It is estimated that "active, reclaimed, or abandoned mountaintop mines cover ~7% of Central Appalachia" with 6.4 km^2 rock displaced in an 11,500 km^2 region (Ross et al., 2016).

There is regulation of coal mining in the USA by the US Department of the Interior's Office of Surface Mining Reclamation and Enforcement under the Surface Mining Control and Reclamation Act 1977. The industry is required to have reclamation programs for mine sites bringing them to the approximate original contours with revegetation. However, this is not occurring with the land 40% flatter with consequent effects on waterways (Ross et al., 2016). Moreover, contamination, for instance with selenium, has been documented (Ross et al., 2016). Whether the timber has been harvested prior to surface mining, reclamation vegetation will initially be grassland such that the forest habitat is lost and grassland habitat is gained. This will affect forest and grassland species. Reclamation of forests obviously takes much more time. There is a real question as to whether the regulations are working as coal companies can avoid reclamation through bankruptcies (New York Times Editorial Board, 2016).

John Grisham's "Gray Mountain" (2014)

The impact of coal mining in Appalachia is the back-story in "Gray Mountain" (2014) by John Grisham.

19.4.3 Wind Turbines and Habitat Loss/Degradation

The number and size of wind farms has been growing with the shift to alternative renewable energy. This laudable goal is not without impacts. There is strong evidence of sizable mortalities of birds. However, the published research reports have large differences in mortalities. One study in 2009 reported 20,000 birds per year killed by wind power and contrasted this with 330,000 killed by nuclear plants and 14 million killed by fossil fuel power plants (Sovacol, 2013). In other studies, estimates of the number of birds killed by wind turbines in the USA in 2012 were an order of magnitude higher ranging from 234,000 to 573,000 birds (Loss et al., 2013;

Smallwood, 2013). These estimates may not be independent or even unbiased. In a detailed comprehensive and independent analysis, it was estimated that bird mortalities were 0.026 per turbine per day and 0.017 per megawatt per day with half the mortalities in nocturnally migrating passerines (Grodsky et al., 2013). Based on there being 49,000 wind turbines in the USA (American Wind Energy Association, 2016), bird fatalities can be extrapolated to a total of 427,000. The impact of turbines will vary with the species of bird. It is thought that raptors (diurnal carnivorous birds) are particularly sensitive to wind turbines (reviewed by de Lucas et al., 2012; Hoover and Morrison, 2005; Madders and Whitfield, 2006). Smallwood (2013) estimated raptor fatalities in the USA at 83,000 in 2012. In Spain, there is high mortality of Griffon vultures in some wind farms (de Lucas et al., 2008) with a mortality of 0.088 per turbine per year (de Lucas et al., 2012). There is also evidence for the reduced reproductive success of white-tailed eagles (*Haliaeetus albicilla*) with territories within 500 m of wind turbines (Dahl et al., 2012).

Estimates of bat fatalities from wind turbines in the USA are 888,000 bats in 2012 (Smallwood, 2013). No differences were observed whether the turbines were lit or unlit at night (Arnett et al., 2008) or with the rotor diameter, but there was an exponential increase with greater turbine tower height (Barclay et al., 2007).

19.4.4 High-Voltage Electrical Transmission Lines and Possible Habitat Degradation

There is little evidence that the presence of high-voltage electrical transmission lines has any effect on the degradation of the environment and may even have positive effects. There were no discernible effects of the electrical and magnetic fields close to these lines on the biology of kestrels (Costantini et al., 2007; Dell'Omo et al., 2009). Parameters measured included growth rate, body condition score, leukocyte counts, and fledging success together with circulating concentrations of melatonin, carotenoids, reactive oxygen metabolites, and antioxidant capacity (Costantini et al., 2007; Dell'Omo et al., 2009). A positive effect of high voltage electrical transmission lines is that Eurasian kestrels (*Falco tinnunculus*) nest in high-voltage transmission tower/pylons.

19.5 IMPACT OF MINING

19.5.1 Overview of Impact of Mining

Mining and the associated processing of ore can lead to habitat loss irrespective of whether it is surface (strip or "mountain top removal") or subsurface (underground and deep underground) mining. Habitat is lost with the construction of access roads and railroads. Moreover, habitat is lost either permanently or temporarily (with reclamation programs) with surface mining (USFWS, 1983) (Section 19.4.2). Associated with mining are the following problems with waterways and lakes:

- shifts in pH (either acidification or alkalinization)
- silting
- increased turbidity reducing photosynthesis
- metals and other toxicants

Mining can degrade or destroy habitats particularly when environmental regulation is lacking or not adequately enforced (Fig. 19.5). Examples of the effects of mining are given next.

19.5.2 Gold Mining

Gold mining can cause considerable damage to the environment. For example, based on satellite data, gold mining was destroying tropical forests in Peru at the rate of 2166 ha year^{-1} before 2008 but increased to 6145 ha year^{-1} between 2008 and 2012 with over a total of 50,000 ha destroyed (Asner et al., 2013). Gold is extracted from high-grade ores using cyanide.

FIGURE 19.5 **Environmental impact of copper mining destroying habitats.** *Source: Image from Shutterstock.*

Processing of low-grade ore can employ alkaline water containing sodium cyanide. After the extraction of the valuable metal, the tailings or leach residues remain together with liquids used in the extraction. Both of these processes result in retention ponds containing tailings, up to 150 ha, with high concentrations of cyanide. These ponds per se can be hazardous to wildlife especially migratory birds (Donato et al., 2007). Moreover, there is the potential of leakage into groundwater or surface waters or massive loss during flooding affecting the entire watershed (reviewed by Eisler and Wiemeyer, 2004). This is problematic for freshwater animal life. Freshwater fish are sensitive to hydrogen cyanide with the median lethal concentration ranging from 57 to 191 ppb but with juvenile fish more sensitive particularly at low temperatures (Smith et al., 1978). There are multiple other potential toxicants from gold mining. These include arsenic, boron, copper, fluoride, mercury, and zinc; these being released during extraction and processing (reviewed by Eisler and Wiemeyer, 2004). There are elevated arsenic concentrations in water, grasses, invertebrates, and so on, near gold mining operations (reviewed by Eisler, 2004a,b). Gold is also recovered using mercury in gold amalgamation (the formation of an alloy of gold and mercury). This can lead to environmental contamination with elevated concentrations of mercury in sediments, phytoplankton, zooplankton, fish, and human samples from areas close to gold mines in developing countries, for instance, in Latin America (Marrugo-Negrete et al., 2008) and historic gold mines, for example, in the USA (reviewed by Eisler, 2004b). The environmental impact of the pollutant mercury is magnified as it bioaccumulates along food chains and hence is a particular problem for marine mammals and birds feeding in estuaries (Costa et al., 2009).

19.6 INFLUENCES OF CLIMATE CHANGE ON HABITAT LOSS

19.6.1 Overview

Climate change is considered in Section 18.4 focusing on the causes of climate change and the role of livestock and poultry as sources of greenhouse gases. This section will discuss examples of the habitat-degrading effects of climate change. The examples will be on the impacts on polar regions and oceans.

19.6.2 Arctic Ice Cover Reductions

The extent and thickness of polar sea ice cover is declining (Comiso et al., 2016; IPCC, 2016) particularly the lowest extent of the cover in late summer (Table 19.6).

TABLE 19.6 Nadir of Sea Ice Extent (Late Summer)

Year	Nadir of sea ice extent in million km^2
1979–88	7.5
1989–98	7.1
1999–2008	6.0
2007	4.3
2012	3.1

Data calculated from Comiso, J.C., Parkinson, C.L., Markus, T. Cavalieri, D.J., Gersten, R., 2016. Current state of sea ice. cover. Cryosphere Science Research Portal. NASA. Available from: http://neptune.gsfc.nasa.gov/csb/index.php?section=234

19.6.3 Impact of Changes in Polar Ice on Polar Bears

Polar bear (*Ursus maritimus*) populations are decreasing along with increasing ice-free days (Hunter et al., 2010).

There has been a decline in sightings of polar bears (Fig. 19.6) in the southern Beaufort Sea region (north of continental Canada) (Bromaghin et al., 2015). Changes in polar bear numbers in the southern Beaufort Sea region are summarized in Fig. 19.7 with the number declining (P = 0.01; adjusted R^2 = 0.560; slope = −88.4) (calculated from data by Bromaghin et al., 2015).

FIGURE 19.6 **Polar bears in Beaufort Sea, Alaska with the habitat degrading due to climate age.** *Source: Courtesy NOAA; Photo by Kelley Elliott.*

Polar bears exhibit changes in behavior and foods consumed to combat the lack of sea ice cover and access to the usual prey species, the primary food of polar bears being ringed (*Pusa hispida*) and bearded (*Erignathus barbatus*) seals. In a study of the Hudson Strait–Northern Hudson Bay Narrows, a low-latitude region of the Canadian Arctic, it was found that polar bear incursions to East Bay Island increased sevenfold from 1997–2001 to 2008–2012 (Iverson et al., 2014). During this time, sea ice coverage was declining (Iverson et al., 2014). Polar bears consumed eggs and destroyed nests of northern common eiders (*Somateria mollissima borealis*) and thick-billed murres (*Uria lomvia*) (Iverson et al., 2014). Parenthetically, polar bears eat garbage when it is available (Lunn and Stirling, 1985). Therefore, human incursions into polar bear-inhabited regions have

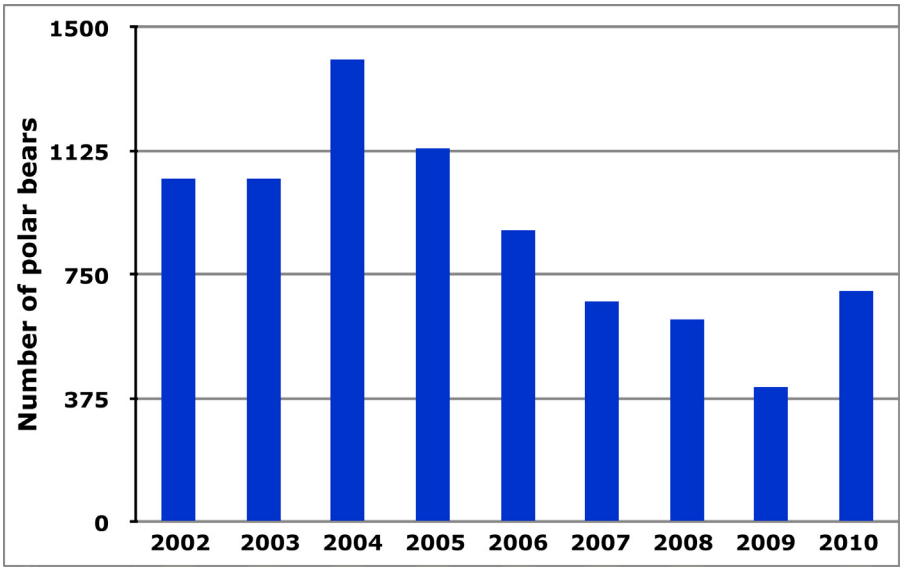

FIGURE 19.7 **Number of polar bears in the southern Beaufort Sea region (north of continental Canada).** *Source: Data from Bromaghin, J.F., McDonald, T.L., Stirling, I., Derocher, A.E., Richardson, E.S., Regehr, E.V., Douglas, D.C., Durner, G.M., Atwood, T., Amstrup, S.C., 2015. Polar bear population dynamics in the southern Beaufort Sea during a period of sea ice decline. Ecol. Appl. 25, 634–651.*

multiple effects: eliminating a pristine environment while augmenting the food availability to polar bears.

19.6.4 Impact of Changes in Polar Ice on Birds

The Arctic Circle lies at 66.3 81 degree N. Since 2005, the Franz Josef Land archipelago (81 degree N and part of the Russian Arctic National Park) has been virtually free of sea ice in the summer.

Little auks (*Alle alle*) lost access to sea ice–associated prey. However, they exhibited plasticity of behavior switching from foraging at distant ice edges to much closer glacier meltwater fronts (Grémillet et al., 2015). There were also no differences in growth rate (Grémillet et al., 2015). In contrast, the survival of adult little auk decreased with the North Atlantic Oscillation index and local summer sea surface temperature in a study of little auk breeding sites close to the West Spitsbergen Current (78 degree N) (Hovinen et al., 2014). This was probably mediated by "change in food quality and/or availability" (Hovinen et al., 2014).

19.6.5 Climate Change and Oceans

19.6.5.1 Ocean Temperatures

Rising ocean temperatures, due to global warming/climate change, per se are likely to have negative and positive effects. For instance, increased ocean temperatures have been reported to increase the growth rate of salmon smolts. However, salmon eggs are susceptible to environmental challenges with a low tolerance to differing temperatures (Elliott and Elliott, 2010). There is decreased oxygen concentration in subsurface oceans due to increased temperatures (reviewed by Doney et al., 2007). Moreover, with increasing ocean temperatures, there are increases in severe weather, for example, hurricanes resulting in mortality of wildlife (Church et al., 2013).

19.6.5.2 Sea Level

Sea level increased due to thermal expansion together with meltwater from polar ice caps (Antarctic and Greenland) and glaciers at a rate of 1.7 mm yr^{-1} between 1900 and 2009 (Church and White, 2011) with a projected further increase of 0.26–0.53 m by 2100 (Church et al., 2013). The rate of sea level increase appears to be accelerating (Haigh et al., 2014). Sea level increases threaten some habitats including the Indo-Pacific mangrove forests (Lovelock et al., 2015) and atolls (Storlazzi et al., 2015).

19.6.5.3 Ocean Carbon Dioxide

A major sink for carbon dioxide from the combustion of fossil fuels is the world's oceans taking up 40% of the anthropogenic-derived carbon dioxide since the beginning of the industrial revolution (Reid et al., 2009). With increasing carbon dioxide dissolved in the oceans (Chapter 18), there are also changes in the pH of these waters. The ocean pH has decreased by 0.1 since prior to the industrial revolution (Royal Society, 2005). The ocean pH is projected to continue to decline by a further 0.3–0.4 by 2100 due to anthropomorphic CO_2 emissions (Royal Society, 2005). Warming of arctic waters may potentially release the powerful greenhouse gas, methane, from hydrates on the ocean floor.

The driving forces for ocean acidification include the following (Doney et al., 2007):

- There is increased concentration of dissolved carbon dioxide (total CO_2 from fossil fuels in the oceans globally = 530 billion tons or gigatons with 1 million tons being added per hour).
- Fossil fuel combustion is associated with sulfur dioxide (0.8 Tmole reactive sulfur per year) and NO_x (2.7 Tmole nitrogen per year) emissions and hence sulfite/sulfate and nitrite/nitrate deposition in oceans particularly in coastal regions.

Ocean pH and Calcium

Changes in the pH in the oceans influence the saturation state for calcium carbonate (Feely et al., 2004). Increased carbon dioxide results in decreased ocean carbonate following this equation:

$$CO_2 + CO_3^{=} + H_2O \rightarrow 2HCO_3^{-}$$

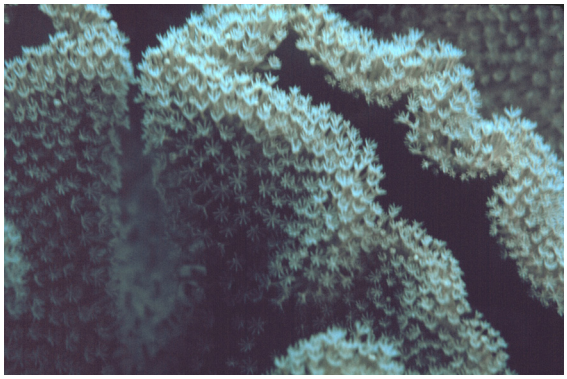

FIGURE 19.8 **Coral reef.** *Source: Courtesy of NOAA; Photo by Dr. James P. McVey.*

Increased carbon dioxide partial pressure in oceans is an example of habitat degradation. This has adverse effects on marine animals. Pteropods are particularly vulnerable to ocean pH as their shells are fragile and composed of aragonite, a relatively soluble form of calcium carbonate (Seibel et al., 2012). The decreased pH of the oceans is projected to impact the stability of the external calcium carbonate skeletons in such marine organisms as corals and some plankton (Orr et al., 2005). It has been demonstrated experimentally that increasing the concentrations of carbon dioxide reduces the thickness of aragonite (calcium carbonate, $CaCO_3$) in the shells of mussels (Fitzer et al., 2015). Elevated carbon dioxide partial pressure depresses oxygen consumption in the pteropod *Limacina helicina* (Seibel et al., 2012).

There is great concern about the impact of ocean acidification on coral reefs (Hoegh-Guldberg et al., 2007) (Figs. 19.8 and 19.9). Coral cover decreased by >85% between 1996 and 2002 in four marine reserves in the Tamane Puli Conservation Area, Papua New Guinea (Jones et al., 2004). Calcification rates of *Porites* corals decreased 14.2% in the Great Barrier Reef between 1990 and 2008 (De'ath et al., 2009). This change was unprecedented in the last 400 years (De'ath et al., 2009). The rates of coral growth are declining and with a constant rate of erosion, corals may be close to a "tipping point" (Hoegh-Guldberg et al., 2007). Coral bleaching is increasingly seen as a symptom of adverse

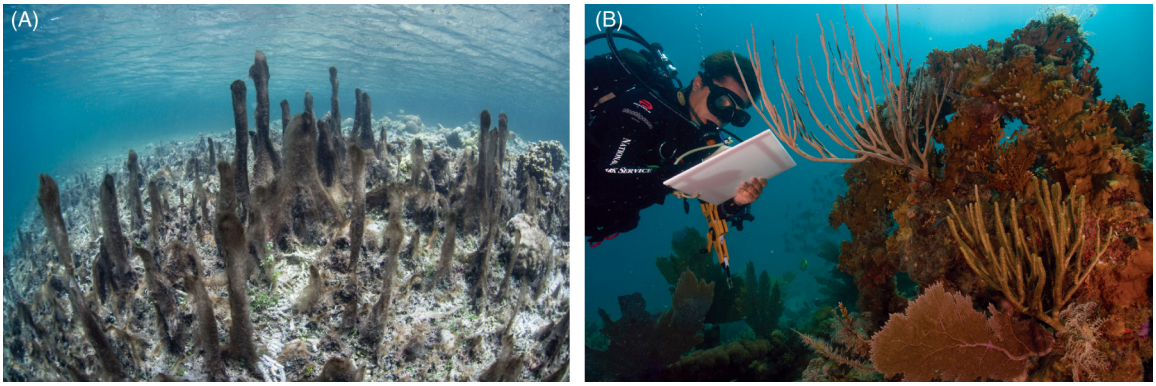

FIGURE 19.9 **Algae growing on destroyed coral in the western Pacific Ocean as the result of climate change.** *Source: Image from Shutterstock.*

health and again it is thought to be anthropogenic (Glynn, 1996).

There are other effects of changes in ocean pH. The ability of clown fish larvae to discriminate by olfaction and avoid water from containing cues from their parents is lost in acidified waters (Munday et al., 2009). Ocean acidification to the level projected for 2100 was simulated by the addition of carbon dioxide (Munday et al., 2009). Acidification impairs settlement of larval fish (Rossi et al., 2015). In studies where carbon dioxide partial pressures are elevated due to natural volcanic vents there is reduced orientation ability of fish larvae due to altered hearing (Rossi et al., 2016). Perhaps surprisingly, lower pH in oceans influences the transmission of sound waves and thereby increases the noise in the ocean (Hester et al., 2008).

19.6.5.4 Adapting to the Effects of Climate Change and Oceans

There is evidence that animals can be resilient or adapt to degraded habitation. For instance, Antarctic sea urchin shells with high-magnesium calcite skeleton appear to be resilient to decreased oceanic pH down decreased carbonate ions (Collard et al., 2015). Climate change-induced changes may also have positive effects with, for instance, evidence that it is responsible for the increase in the population of Chinstrap penguins (Sun and Xie, 2001). Moreover, sea urchins not only survive but also thrive when carbon dioxide partial pressures are elevated close to volcanic vents (Uthicke et al., 2016). Also, there can be shifts in food supply.

19.7 OTHER HUMAN IMPACTS OF HABITATS

There are other effects of human activity from habitat loss, degradation, or fragmentation. Examples of these include the following:

- Dams for hydroelectric power and flood control negatively influence breeding grounds for fish.
- Pollutants degrading habitats. For instance, polychlorinated biphenyls bioaccumulate in marine mammals, such as harbor seals (*Phoca vitulina*), which are at the top of the food chain (Park et al., 2009).
- The ecologically fragile Florida Everglades (Fig. 19.10) is vulnerable to the effects of climate change (Catano et al., 2015) and the invasive species, the Burmese python (discussed in Section 17.4.1).
- The development of a man-made lake and opening of a habitat for human recreation was accompanied by declines in populations of North American wood turtles (*Clemmys insculpta*) (Garber and Burger, 1995).
- Human activity can influence the number of animals by causing disturbance. For instance, the number of sharks was reported to be lower in areas of the Great Barrier Reef that were fished compared with "no entry" or "no take" areas (McCook et al., 2010). It is assumed that this reflects shifts in the number of fish eaten by the sharks (McCook et al., 2010).

19.8 CONCLUSIONS

Clearly, human activity is leading to habitat loss, habitat degradation, and habitat fragmentation. It is questioned whether a tipping point has already been reached or whether we are close to an anthropogenic major extinction event. It is also recognized that even when policymakers base decisions on "good science," concerns about economic development and poverty alleviation may outweigh the importance of healthy ecosystems. The effects of damage to habitats need to be part of the equation. The issue is not about no regulation or no growth but smart regulation with sustainability or smart growth addressing poverty, low quality of life, poor nutrition, and diseases.

FIGURE 19.10 **The Everglades in Florida (USA) are ecologically fragile with habitats is danger of destruction.** (A) Wild turkeys in the Everglades National Park. (B) Alligator in the Everglades National Park. (C) Egret in the Everglades National Park. (D) Seagrass meadows on the western edge of the Everglades providing nutrition to many animal species. *Source: Part A and C: Courtesy US National Parks Service; part B: Courtesy US National Parks Service; photo by Karen Battle-Sanborn; part D: Courtesy US National Parks Service; photo by Justin Campbell.*

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ANIMAL WASTE

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Humans have consumed animal-sourced food products since the beginning of mankind. These products provide high nutrient density and nutritional quality food options for consumers and currently represent approximately 1/6 and 1/3 of the food energy and food protein, respectively, for humans on a global basis (Bradford, 1999; CAST, 2006). The estimated current annual global production of beef, pork, and poultry meat is approximately 250 metric tons (mt) (USDA, 2015). The societal impacts of producing this amount of food animal products is profound considering that global meat consumption is projected to grow by an annual average of 1.4%, which will result in an additional annual global consumption of approximately 50 million mt by 2024 (OECD/FAO, 2015). These statistics do not include the global egg industry, which represents approximately 5 billion egg-laying hens (International Egg Commission, 2016). In the United States alone approximately 9 billion broiler chickens, 33 million cattle, 240 million turkeys, 2 million sheep and lambs, and 110 million hogs are processed annually (North American Meat Institute, 2016); the United States dairy cattle inventory represents an additional estimated 9 million heads of animals (USDA-ERS, 2016). Using poultry and pork as examples, the United States percent of global production is only approximately 26 and 10%, respectively. The overarching topic of food animals and human society is critically important from environmental, human health, and food security perspectives—it is essential to consider the issues, challenges, and opportunities associated with this topic from a global, not regional, perspective.

The beneficial nutritional contributions of food animals are well recognized and documented in the scientific literature; however coproducts (traditionally referred to as animal waste) associated with the production of these animals can have beneficial, as well as detrimental impacts on society. For a global perspective consider that animal waste manure dry matter (mass after water content is removed) from food animals housed only in the United States is estimated to annually exceed 335 mt (USDA-ARS, 2005). In the United States and elsewhere the management of animal waste in the 21st century continues to predominately follow a model of land application of these coproducts proximate to where the animals are produced—mostly for fertilizer value for targeted crop(s) production. Approved, permitted, and recommended best management practices for land application of animal waste has historically been beneficial and, arguably, is not currently presenting significant environmental or human health threats in many areas utilizing these practices. However, these traditional practices for managing animal waste are unlikely to remain sustainable for many reasons. With the projected increase in global population of humans over the next few decades there will be less farmland available. In addition the future competition between crop and animal agriculture, as well as nonagricultural entities for water resources will be intense (many animal waste management practices currently utilize large volumes of water). Other factors that will impact the future of animal waste management practices include consumer attitudes regarding environmental issues associated with food animal production. These issues, real or perceived, are currently impacting global livestock and poultry production and will only intensify in the future.

An objective assessment of available evidence shows that issues associated with traditional animal waste management practices are geographically specific and include potential discharge of nutrients and pathogenic bacteria to surface and groundwater, and aerial emissions of volatile organic compounds, ammonia, hydrogen sulfide, dust, pathogens, and odor. The consequences of improperly managed animal waste include environmental and human health impacts from elevated concentrations of nitrates, organic matter, sediments, pathogens, heavy metals, hormones, and antibiotics (EPA, 2016). Of particular concern is the emission of gaseous ammonia from animal production housing facilities and associated land application animal waste management practices (EPA, 2004). Atmospheric ammonia (and ammonium) can redeposit via wet and dry deposition processes into aquatic ecosystems that may be nitrogen sensitive resulting in eutrophication (Jones et al., 2013). Ammonia that returns from the atmosphere into certain terrestrial environments may be metabolized by soil bacteria resulting in the release of nitrous oxide, which can potentially impact climate change. Perhaps more concerning is that under certain conditions in the atmosphere ammonia can be converted to ammonium compounds, which comprise a fraction of fine particulate matter (Kwok et al., 2013). Elevated levels of certain fine particulates may have adverse human health impacts (Pope et al., 2009). As with many nonagricultural industrial processes that have benefited society over the past century, animal production agriculture results in externalities due to the resulting environmental emissions to soil, water, and air media. The future of global food security will be dependent upon fair and just solutions regarding the fate and management of animal waste. This will require implementation of proven innovative technology, as well as the development of new food animal agricultural production and management practices.

In summary, it is recognized that nutrients in animal manures can be a valuable source of organic plant nutrients which, if properly managed, reduces the need for chemical fertilizer—a practice that historically and currently is resulting in improved soil conditions and crop production in many parts of the world. It is also recognized that carbon compounds in animal manures can be a source of energy—many areas of the world are benefiting from the utilization of this renewable energy resource. Conceptually, animal waste resources offer holistic animal production opportunities with appropriate technology applications (Shih, 2015). However, economic constraints limit the widespread application of proven technologies in many areas. Policy change, as well as technology development will be required for widespread implementation of new animal agriculture waste management practices (Williams, 2009). A collective global governance and policy initiative providing reasonable incentives for implementing appropriate technologies by providing technical and financial resources can significantly impact environmental and societal issues associated with food animal production agriculture, as well as maintaining the long-term sustainability of this economically important agricultural sector.

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WILDLIFE AND ENVIRONMENTAL POLLUTION

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The intentional use of poisons can be traced to ancient man (venoms and plant extracts used in hunting and warfare) and the cradle of civilization (Ebers Papyrus, an Egyptian medical document, describes hemlock, aconite, and lead), with the first hint of potential pollution effects in animals associated with the ancient Greek maxim "a bad crow lays a bad egg" (Rattner, 2009). While the term pollution has a broad meaning, encompassing release of naturally occurring or anthropogenic agents (e.g., chemical, pathogens, light, noise, and temperature) that potentially cause harm to the environment, in the present context its use will be limited to chemical stressors. In the 13th century, complaints about smoke from burning of coal in London resulted in a royal proclamation by Edward I prohibiting the use of "sea-coal" due to its annoyance and the concern over "injury of their body health" to man (Mosley, 2014). However, awareness of pollution and wildlife (viz. amphibians, reptiles, birds, and mammals) did not emerge until the late 19th century when popular press reports of pheasant (*Phasianus colchicus*) and waterfowl mortality were related to ingestion of spent lead shot. Once recognized, lead poisoning was considered a common occurrence in waterfowl and other species of birds through the 20th century to the present time. Despite extensive research and regulatory and conservation efforts, intoxication and poisoning associated with accidental ingestion of spent lead ammunition persists today in charismatic species, such as the bald eagle (*Haliaeetus leucocephalus*) and California condor (*Gymnogyps*

californianus). Wildlife toxicology is the study of exposure and potential effects of chemical agents (e.g., agrichemicals, industrial compounds, pharmaceuticals, and personal care products) and anthropogenic processes (e.g., irrigation, mining, capture and transport of fossil fuel, and energy generation) on the individual, population, community, and ecosystem. The hazard of spent lead ammunition typifies the discipline. As a component of ecotoxicology, wildlife toxicology has been shaped by chemical use and misuse, ecological mishaps and other catastrophic events, and occasionally by deliberate hypothesis testing, discovery, and accretion of basic knowledge common to the advancement of most other fields of science (Rattner, 2009).

By the 1930s, about 30 pesticides (e.g., pyrethrum, nicotine, arsenicals, mercurial fungicides, and dinitro-*ortho*-cresol herbicide) were in use in the United States, United Kingdom, and elsewhere (Sheail, 1985). Aerial application was commonplace, and potential adverse effects on nontarget wildlife were acknowledged at the Third North American Wildlife Conference (Strong, 1938). The birth of the synthetic pesticide era, including the discovery of the insecticidal properties of DDT (1939), held great promise for agriculture and human health, although premonitions of damage to wildlife appeared in the widely read Atlantic Monthly in 1945 (Wigglesworth, 1945). By the close of the decade, DDT field studies documented harm to passerines (George and Stickel, 1949; Hotchkiss and Pough, 1946; Robbins and Stewart, 1949), and by the mid-1950s organochlorine and organophosphorus insecticides (aldrin, dieldrin, schradan, and parathion) used as sprays and seed-dressings were linked to mortality events in game birds and mammals (Sheail, 1985). Rachel Carson's *Silent Spring* (Carson, 1962) described in lay terms a multitude of pesticide effects (e.g., ecological imbalances, persistence, resistance, and loss of wildlife), which triggered discussion and debate at all levels of society, and kick-started the environmental movement, new government agencies, research and monitoring programs, and regulatory legislation. Population declines of some fish-eating and raptorial birds, and later other species (e.g., bats), were of great concern. Following the discovery of many damaged peregrine falcon (*Falco peregrinus*) eggs, Moore and Ratcliffe (1962) and others detected organochlorine pesticide residues in eggs, and linked the appearance of eggshell thinning with DDT use (Hickey and Anderson, 1968; Ratcliffe, 1967). Organophosphorus and carbamate insecticides gradually replaced organochlorine pesticides, and while less persistent and seemingly safer, ingestion of pesticide granules and exposed prey caused bird mortality (Grue et al., 1983; Mineau, 1991). Some of the "die-off" events were large (20,000 Swainson's hawks, *Buteo swainsoni,* in Argentina; Hooper et al., 2003), yet subtle effects on food webs may have indirectly had more profound effects on wildlife. While some of these acetylcholinesterase-inhibiting insecticides are still in use, other classes of pesticides (e.g., pyrethroids and neonicotinoids) began to dominate the marketplace. The neonicotinoids, hypothesized by some to cause bee colony collapse (as yet to be rigorously proven, see Fairbrother et al., 2014) may affect granivorous birds consuming pesticide-coated seeds, and likely indirectly affect other wildlife species by reducing invertebrate prey abundance (Gibbons et al., 2015). Despite extensive requirements for their use and registration, it remains difficult to completely characterize exposure, metabolism, adverse effects, and toxicity thresholds of pesticides to provide the means to protect the diversity of wildlife.

Production and large-scale use of the polychlorinated biphenyls (PCBs; 209 congeners) as insulating chemicals, lubricants, and flame retardants (1929–1977 in the US), and their transfer through water, air, and the food chain has resulted in global exposure and often adverse effects in wildlife.

Reproductive problems in waterbirds related to some of the coplanar PCB congeners and other dioxin-like compounds led to the description of the Great Lakes Embryo Mortality, Edema and Deformity Syndrome that met various epidemiological criteria (Gilbertson et al., 1991). Similarly, wasting, edema, immunotoxicity, and reproductive effects were described in mink and marine mammals. While the manufacture, processing, and distribution of PCBs ended in 1979, and remediation efforts have been underway for decades, PCB congeners are routinely detected, and depending on exposure dose, are accompanied by toxic effects in wildlife. Replacement flame retardants, including polybrominated diphenyl ethers, are less toxic but still bioaccumulate (Chen and Hale, 2010), and their use has been phased out in favor of short-lived alternative flame retardants.

Scientific workshops on chemically induced alterations of endocrine and reproductive function, in concert with the publication of *Our Stolen Future* (Colborn et al., 1996), resulted in extensive efforts to identify endocrine disruptive compounds (EDCs) through traditional toxicity tests, high-throughput screening assays, and field studies. While the adage *the dose makes the poison* has been known for centuries (credited to Paracelsus, the founder of toxicology), *the timing* (sic. of exposure) *makes the poison* has been the mantra of EDCs (Gore et al., 2014). Although much has been learned from controlled exposure studies, EDC effects in free-ranging wildlife have been difficult to document. Notable examples include atrazine-induced hermaphroditism in America leopard frogs (*Rana pipiens*) (Hayes et al., 2003) and feminization of alligators (*Alligator mississippiensis*) potentially caused by organochlorine pesticides (Woodward et al., 2011). The complexity of endocrine-mediated effects and interactions with other physiological systems and behavior is challenging to study in free-ranging biota, and thus evidence for causal linkages between EDCs and effects in wildlife remain incomplete and controversial (Hotchkiss et al., 2008; Murphy et al., 2006).

While some pharmaceuticals and personal care products may be classified as EDCs, drug effects on wildlife can also be of a more direct toxic nature. One such example is the nonsteroidal antiinflammatory drug (NSAID) diclofenac used for the treatment of inflammation, fever, and pain in domestic livestock. Through a remarkable forensic investigation, it was determined that diclofenac was unintentionally ingested by Old World vultures (*Gyps* species) scavenging on carcasses of livestock that had received diclofenac therapeutically shortly before death (Oaks and Watson, 2011; Oaks et al., 2004). Diclofenac is a potent inhibitor of cyclooxygenase-2 and prostaglandin synthetase, and differential sensitivity among avian species is a hallmark of NSAIDs. Mechanistically, it is hypothesized to impair smooth muscle control of the renal portal valve, shunting blood from the renal cortex, causing ischemia, necrosis, visceral gout, and death within a matter of days. Diclofenac appears to have been the principal cause of the unprecedented population crash of *Gyps* vultures in India, Pakistan, and Nepal, and is the only well-documented instance of a veterinary drug resulting in population-level effects and the endangerment of a wildlife species. The decline had implications for humans in the area, including the cost of livestock carcass disposal, cultural and religious impacts, and health implications including the increase in dog populations, which are the principal source of rabies in India (Markandya et al., 2008). While diclofenac use in livestock has been banned in India, and a safe replacement NSAID (meloxicam) is available, its use continues in much of the range of *Gyps* vultures.

Anthropogenic processes, such as irrigation and mining can have profound effects on wildlife. In the 1970s, the potential use of agricultural drainage water for wetland management was

being considered in parts of California (Ohlendorf, 2011). In the San Joaquin Valley, natural drainage was inadequate for crop production because saline water accumulated in the plant root zone, but could be ameliorated through subsurface drain tiles to collect and convey saline groundwater from fields. The US Bureau of Reclamation constructed a canal system to dispose of this drainwater; however, the project was not completed as planned and the canal terminus ended at the Kesterson National Wildlife Refuge rather than San Francisco Bay. At this Refuge, irrigation drainwater was impounded in shallow ponds as a beneficial use for wildlife. Monitoring studies revealed that the concentrations of most metals, metalloids, and organochlorine pesticides in mosquito fish (*Gambusia affinis*) from sites receiving drainwater and an untreated reference area were similar; however, levels of selenium in mosquito fish from the drainwater sites were nearly an order of magnitude greater. Field and laboratory studies quickly demonstrated that selenium concentrations in the wildlife food chain at the Refuge were causing epidemic-like problems for waterbirds (up to 40% of nests contained deformed or dead embryos or chicks), and similar selenium-related toxicity in wildlife was found to be widespread in many parts of the western United States. Remediation and management of this situation remains difficult, involving a combination of actions (e.g., source control, drainage reuse, evaporation systems, land retirement, and controlled discharge) (Ohlendorf, 2011).

Mining and smelting of lead, silver, and gold in northern Idaho began in the 1880s and resulted in the deposition of approximately 75 million tons of metal-enriched sediment in Lake Coeur d'Alene (Henny, 2003). Waterfowl collected from this area in 1955 contained high levels of lead, zinc, and copper in their tissues, which was independent of ingestion of spent lead shot. In the decades that followed, annual die-offs of tundra swans (*Cygnus columbianus*) and waterfowl continued. Through a series of field and laboratory studies examining lead, other metals, and biomarker responses (e.g., inhibition of δ-aminolevulinic acid dehydratase activity, blood protoporphyrin concentration, and presence of renal inclusion bodies), the cause of mortality was linked to the ingestion of "mining-related lead" in sediment and biota rather than the ingestion of spent lead shot from hunting (Beyer et al., 2000; Henny, 2003). These findings resulted in one of the largest natural resource damage assessments of the US Department of the Interior, and the collection of nearly $80 million from mining polluters that was ultimately used for the recovery of wildlife and habitat (US Fish and Wildlife Service, 2009).

Collection and transport of fossil fuels, and the generation of energy, are an integral part of modern civilization, but at times have had devastating effects on wildlife and their supporting habitat. Often thought of as a contemporary issue, petroleum spills accompanied by the oiling and death of waterbirds and other biota have been documented for nearly a century, dating back to maritime events of the 1920s (Phillips and Lincoln, 1930). Highly publicized events (e.g., *Torrey Canyon*, 1967; *Exxon Valdez*, 1989; Gulf War oil spills, 1991; and Deepwater Horizon, 2010), including striking images of petroleum sheen on water, tar-covered habitat, and oil-drenched birds, have captivated public attention and emotion. Interpretation of observations and recovery following petroleum spill events are often controversial, pitting the presumably responsible party against trustees of the natural resource (Wiens, 1996). Extensive rehabilitation efforts of oiled biota, while noble, have yielded mixed results on postrelease survival (Russell et al., 2003).

The meltdown of the Chernobyl nuclear reactor in 1986 released radiocesium and other radionuclides (~3,000,000 trillion Becquerels) resulting in acute and latent effects in humans, plants, and animals at, and a great distance from, the accident site (Eisler, 2003). A nearby pine forest within

the "exclusion zone" received a dose of >80 Gy (Gy = 100 rad), died, turned ginger-brown in color, and was subsequently dubbed the Red Forest. There was significant mortality of small mammals in this area; however, the following year rodent population seemed normal, possibly due to immigration from less affected areas. Radioactivity in tissues of game birds near Chernobyl and in large mammals (reindeer, *Rangifer tarandus*; moose *Alces alces*) over 1000 km from Chernobyl, exceeded levels safe for human consumption for years. Several teams of investigators examined genetic effects in small mammals from Sweden and Italy, and at Chernobyl, with findings being inconsistent (Deryabina et al., 2015). A detailed study of the voles (*Microtus* sp.) in the exclusion zone revealed considerable genetic variation, which subsequently was demonstrated to be natural variation among four species of voles (Chesser and Baker, 2006). Although Chernobyl was considered a large-scale disaster, ironically, the subsequent reduced human presence has resulted in abundant populations of wildlife in the exclusion zone (Deryabina et al., 2015).

Man's dependence on and interaction with wildlife can be traced to the beginnings of recorded history. In 1990 just before the 20th anniversary of *Earth Day,* the National Wildlife Federation published a special issue of their magazine entitled *What on Earth Are We Doing?* (National Wildlife Federation, 1990). Dramatic photographs and articles entitled *Innocent World of a Toxic Victim*, *Pollution Knows No Boundaries*, and *Your Contribution of Global Warming* highlighted pollution issues affecting the biosphere, some of our accomplishments, and the many challenges that man and wildlife face. In my lifetime I have tracked societal and regulatory activities that have protected wildlife and habitat. Many of us have witnessed "nature's great experiment" with the recovery of the bald eagle, brown pelican (*Pelecanus occidentalis*), osprey (*Pandion haliaetus*), and peregrine falcon (*Falco peregrinus*) populations that were devastated by many chlorinated pesticides. There have been advances in green chemistry to replace chemicals that may be persistent, bioaccumulative, or toxic with compounds that are benign by design. Nonetheless, there remains an ongoing conflict between wildlife and man over the habitat we share, with a significant component related to chemical use, agriculture, demand for natural resources, and the generation of energy. Our approach to these conflicts has become holistic, multidisciplinary, and risk-based. The contemporary One Health Initiative that seeks to attain optimal health for people, animals, and the environment, while new, is not really a new concept. Wildlife has time and again served as man's canary in the coal mine, and we need to protect the canary! Clearly, some pollution threats to wildlife have been resolved, "wicked problems" linger, with the unknown looming on the horizon.

Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the US Government.

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HUMAN–WILDLIFE CONFLICTS

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The degree and scope of human–wildlife conflicts is a matter of perspective. Some see the issue through the lens of how human activities and interests have negatively impacted wildlife populations. Others see the issue through the lens of how wildlife negatively impact humans' economic or health interests. In either case, the passions can run deep. The scenarios for solutions have played out differently across the globe (Distefano, 2005; Madden and McQuinn, 2014). Regardless, the issue transcends economic sectors and development status of any given country and is deeply rooted in the cultural practices of agricultural and natural resource sustainability.

In the United States, the issues surrounding human–wildlife conflicts reached a critical juncture in the late 19th and early 20th centuries. Commercial exploitation of wildlife by market hunters was devastating North American wildlife populations (Organ et al., 2012). Alarmed, conservationists and sportsmen began to formalize wildlife management policies into what is now known as the North American wildlife management model, establishing seven principles for the sustainable harvest of wildlife species for the benefit of human consumption, conservation, and recreation (Tober, 1981). At the same time agricultural interests were concerned about the negative impact wildlife had on crop and livestock production. As a result of these concerns the Division of Economic Ornithology and Mammalogy was established under the Department of Agriculture. This division's early purpose was to study food habits of wildlife and educate the agricultural community as to which wildlife species were beneficial or harmful to their interests, and eventually develop

programs for wildlife control to prevent damage to crops and livestock. Later, the division took on the task for the prevention of transmission of wildlife disease to human health and economic interests (e.g., rabies) (Hawthorne, 2004).

Both efforts have evolved into well-established and regulated systems of administration and management of wildlife in the United States, yet the inherent conflict of what the social carrying capacity of wildlife should be remains in place (Decker and Purdy, 1988). Fundamentally, wildlife valuation is related to direct experience. A homeowner, farmer, rancher, and so on, generally are less tolerant of wildlife and wants more direct intervention if they have experienced damage to their property or commodity than those who have no such negative experience (Young et al., 2015). Similarly, those who value wildlife for recreation or heuristic reasons are more apt to oppose interventionist management strategies (Reiter et al., 1999). These perspectives have led to some interesting policy debates surrounding methods to resolve conflicts, the economics and effectiveness of the methods, and who should bear the costs of damage should it occur given that, at least in the United States, wildlife are held in public trust (Bulte and Rondeau, 2005; Nyhus et al., 2005). As controversial as some government-sponsored programs might be in terms of compensation, mitigation, or control involvement, when individuals experience conflicts with wildlife, and in the absence of sanctioned and monitored regulated assistance or relief, the consequences of individually or privately based remedies usually are not a favorable outcome for wildlife (Mateo-Tomás et al., 2012). That is to say, individual economic interests will prevail.

By 2050 the world will have 10 billion people, resulting in a rapidly declining per capita availability of arable land. One consequence of this fact is that there will be enormous pressure to extract as much from the land to produce food for an increasingly hungry world (Green et al., 2005). The growth of the world's population (Phalan et al., 2011), pressures of land and water management (Tilman et al., 2011), food production (Henle et al., 2008), industrialization (Gates et al., 2014), and increased globalization of trade and transport (Hulme, 2009) have all resulted in newer and more intense manifestations of human–wildlife conflicts requiring urgent attention.

Other high-profile examples serve to illustrate the diversity and complexity of human–wildlife conflicts. Each example put forth is a mix of biological processes and human dimensions issues, which include economics, emotions, and value systems. The approaches to solutions range from technical, regulatory, and broader policy. The best cases of addressing and solving any particular human–wildlife conflict has been the use of data and analyses with integration across science and social disciplines, and it demonstrates how successful a willingness of the partners to engage in dialog can be.

Because carnivores eat livestock, and sometimes harm people, human–carnivore conflicts are emblematic of the longstanding economic and emotional issues encountered in the arena of human–wildlife conflicts. Pastoralists and ranchers are closely tied to their animals through culture and economy, thus the loss through predation is monetary and personal (Dickman, 2010). Historically, extirpation of carnivores was a short- and long-term solution. But changes in landscape and ecosystem recovery efforts and targeted species recovery have renewed traditional conflicts. In Europe, in the absence of predation certain management practices, such as livestock protection using guard dogs was lost; where predation remained, those traditions persisted (Kaczensky, 1999). As society values change, the producer must make the economic and societal management decisions on the cost-effectiveness of different practices, that is, lethal damage management control and/or nonlethal methods (which are usually spatially limited and more costly).

Another high-profile area of research and management is aviation safety (DeVault et al., 2011). Keeping birds out of the flight path of aircraft is uniformly seen as a positive goal. In general, two approaches are used. Hazing is the practice of chasing birds away by sound, visual, or other harassment systems. The effects are short-lived with animals rapidly habituating to the methods. Longer-term solutions generally focus on land management practices in and around airports (DeVault et al., 2013). In this case, the goal is to make habitats unattractive to species of birds that pose the highest risks to aircraft, such as flocking species and species with high body density.

Wildlife disease increasingly has become an issue of human–wildlife conflict (Clark, 2014; Karesh et al., 2005; Kock, 2005). The recent visibility of highly pathogenic avian influenza, its impacts on the global poultry industry, and its wildlife origins are good examples of the large-scale magnitude that the conflict can reach (Clark and Hall, 2006). Yet devising the appropriate risk assessments and farm-side management practices to enhance biosecurity against exposure to wildlife is proving to be a challenge that pits agricultural production against wildlife conservation (Graham et al., 2008).

Invasive species may cause conflict because humans value biodiverse communities and invasive species can negatively impact native ecosystems (Olson, 2006). For example, brown tree snakes have been implicated in the extinction of several species of native birds on Guam (Rodda and Savidge, 2007). Interdiction and control efforts are in part used for natural resource recovery, but primarily focused on containment to prevent further environmental disasters by inadvertent transportation routes. But as in most cases, the difficult decision of when to manage an invasive species often depends on resources and perceived attitudes of the threat (Sharp et al., 2011). Because of the large economic and other resource investments needed for such control programs, there is an imperative to evaluate when an exotic species reaches a threshold where it becomes injurious, hence invasive and thus rises to a situation where action is warranted.

Finally, there is a rich tradition of research on new technical methods and tools used to execute management (Conover, 2001). However, many of these require regulatory approval often not appreciated by researchers and field managers alike. And if manufacture is needed, a business analysis for the scalability to commercial levels, a business plan, intellectual property protection, and technology transfer become filters to getting successful tools into the hands of managers. Few wildlife managers or conservationists are equipped to participate in these arenas. The need for fully integrated partnerships on the research and development and regulatory side are needed. Of course the marketing and success of getting users to accept methodologies is needed as well. The good news is that approaches to methods development have made great strides and there is better integration of a complex network of perspectives, needs, capabilities, and policy than there was a few years ago.

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